

6. Land-Use Change and Forestry

This chapter provides an assessment of the net carbon dioxide (CO₂) flux¹ caused by 1) changes in forest carbon stocks, 2) changes in carbon stocks in urban trees, 3) changes in agricultural soil carbon stocks, and 4) changes in carbon stocks in landfilled yard trimmings. Seven components of forest carbon stocks are analyzed: trees, understory vegetation, forest floor, down dead wood, soils, wood products in use, and landfilled wood products. The estimated CO₂ flux from each of these forest components was derived from U.S. forest inventory data, using methodologies that are consistent with the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). Changes in carbon stocks in urban trees were estimated based on field measurements in ten U.S. cities and data on national urban tree cover, using a methodology consistent with the *Revised 1996 IPCC Guidelines*. Changes in agricultural soil carbon stocks include mineral and organic soil carbon stock changes due to use and management of cropland and grazing land, and emissions of CO₂ due to the application of crushed limestone and dolomite to agricultural soils (i.e., soil liming). The methods in the *Revised 1996 IPCC Guidelines* were used to estimate all three components of changes in agricultural soil carbon stocks. Changes in yard trimming carbon stocks in landfills were estimated using analysis of life-cycle greenhouse gas emissions and sinks associated with solid waste management (EPA 1998). Note that the chapter title “Land-Use Change and Forestry” has been used here to maintain consistency with the IPCC reporting structure for national greenhouse gas inventories; however, the chapter covers land-use activities, in addition to land-use change and forestry activities. Therefore, except in table titles, the term “land use, land-use change, and forestry” will be used in the remainder of this chapter.

Unlike the assessments in other chapters, which are based on annual activity data, the flux estimates in this chapter, with the exception of those from wood products, urban trees, and liming, are based on periodic activity data in the form of forest, land-use, and municipal solid waste surveys. Carbon dioxide fluxes from forest carbon stocks (except the wood product components) and from agricultural soils (except the liming component) are calculated on an average annual basis over five or ten year periods. The resulting annual averages are applied to years between surveys. As a result of this data structure, estimated CO₂ fluxes from forest carbon stocks (except the wood product components) and from agricultural soils (except the liming component) are constant over multi-year intervals, with large discontinuities between intervals. For the landfilled yard trimmings, periodic solid waste survey data were interpolated so that annual storage estimates could be derived. In addition, because the most recent national forest, land-use, and municipal solid waste surveys were completed for the year 1997, the estimates of CO₂ flux from forests, agricultural soils, and landfilled yard trimmings are based in part on modeled projections. Carbon dioxide fluxes from urban trees are based on neither annual data nor periodic survey data, but instead is data collected over the decade 1990 through 2000. Therefore, this flux has been applied to the entire time series.

¹ The term “flux” is used here to encompass both emissions of greenhouse gases to the atmosphere, and removal of carbon from the atmosphere. Removal of carbon from the atmosphere is also referred to as “carbon sequestration.”

Table 6-1: Net CO₂ Flux from Land-Use Change and Forestry (Tg CO₂ Eq.)

Sink Category	1990		1995	1996	1997	1998	1999	2000
Forests	(982.7)		(979.0)	(979.0)	(759.0)	(751.7)	(762.7)	(770.0)
Urban Trees	(58.7)		(58.7)	(58.7)	(58.7)	(58.7)	(58.7)	(58.7)
Agricultural Soils	(37.3)		(60.2)	(60.2)	(60.4)	(67.2)	(67.7)	(67.4)
Landfilled Yard Trimmings	(19.1)		(12.2)	(10.2)	(9.5)	(8.3)	(7.3)	(6.4)
Total	(1,097.7)		(1,110.0)	(1,108.1)	(887.5)	(885.9)	(896.4)	(902.5)

Note: Parentheses indicate net sequestration. Totals may not sum due to independent rounding. Lightly shaded areas indicate values based on a combination of historical data and projections. All other values are based on historical data only.

Table 6-2: Net CO₂ Flux from Land-Use Change and Forestry (Tg C)

Sink Category	1990		1995	1996	1997	1998	1999	2000
Forests	(268)		(267)	(267)	(207)	(205)	(208)	(210)
Urban Trees	(16)		(16)	(16)	(16)	(16)	(16)	(16)
Agricultural Soils	(10)		(16)	(16)	(17)	(18)	(19)	(18)
Landfilled Yard Trimmings	(5)		(3)	(3)	(3)	(2)	(2)	(2)
Total	(299)		(303)	(302)	(242)	(242)	(245)	(246)

Note: 1 Tg C = 1 teragram carbon = 1 million metric tons carbon. Parentheses indicate net sequestration. Totals may not sum due to independent rounding. Lightly shaded areas indicate values based on a combination of historical data and projections. All other values are based on historical data only.

Land use, land-use change, and forestry activities in 2000 resulted in a net sequestration of 903 Tg CO₂ Eq. (246 Tg C) (Table 6-1 and Table 6-2). This represents an offset of approximately 15 percent of total U.S. CO₂ emissions. Total land use, land-use change, and forestry net sequestration declined by about 18 percent between 1990 and 2000. This decline was primarily due to a decline in the rate of net carbon accumulation in forest carbon stocks. Annual carbon accumulation in landfilled yard trimmings also slowed over this period, while annual carbon accumulation in agricultural soils increased. As described above, the constant rate of carbon accumulation in urban trees is a reflection of limited underlying data (i.e., this rate represents an average for the decade).

Changes in Forest Carbon Stocks

Carbon in forests can be described as the total of several interrelated carbon storage pools, including:

- Trees (i.e., living trees and standing dead trees, including the roots, stems, branches, and foliage);

- Understory vegetation (i.e., shrubs and bushes, including the roots, stems, branches, and foliage);
- Forest floor (i.e., fine woody debris, tree litter, and humus);
- Down dead wood (i.e., logging residue and other coarse dead wood on the ground, and stumps and roots of stumps); and
- Soil (i.e., organic material in soil).

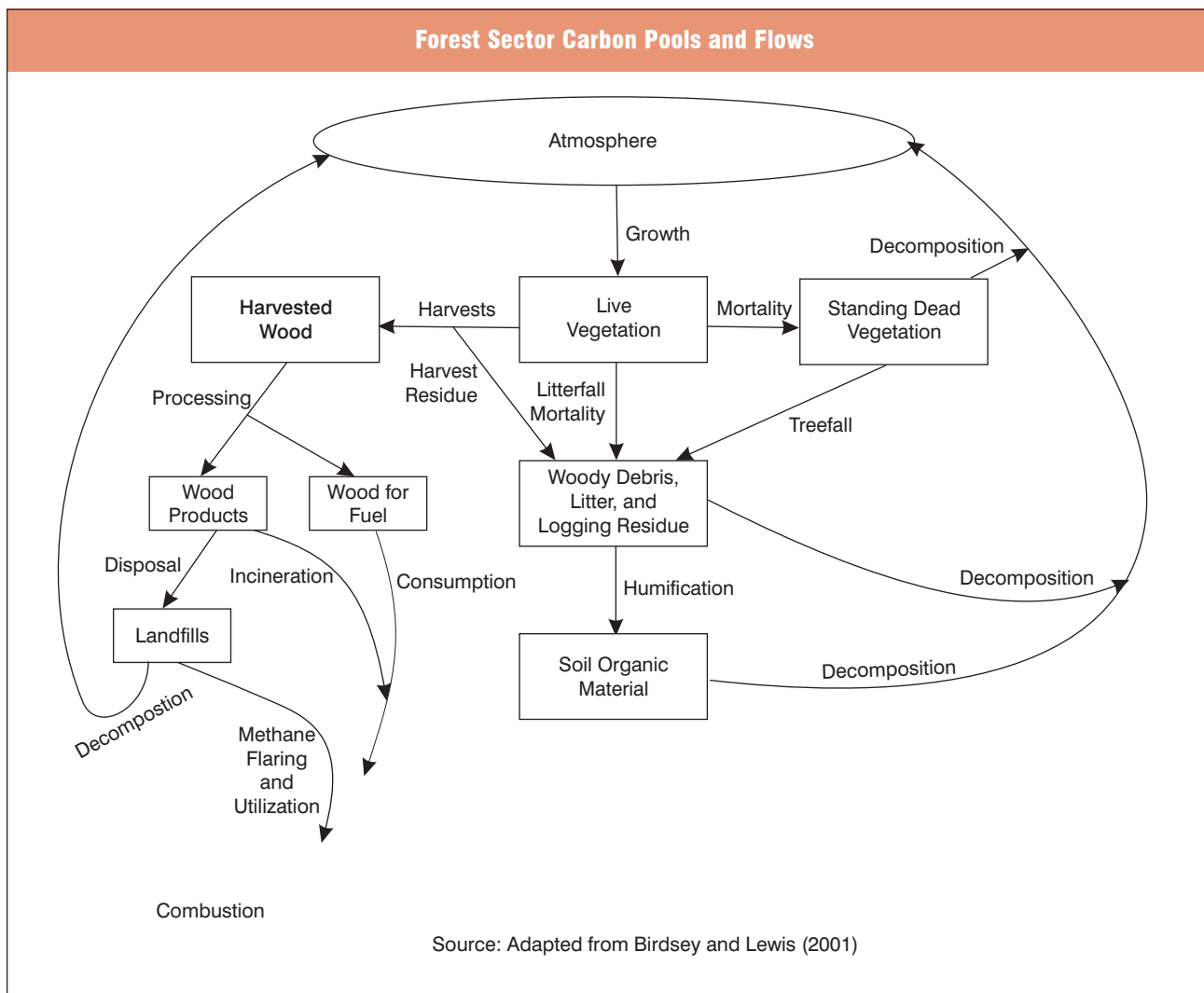
As a result of biological processes in forests (e.g., growth and mortality) and anthropogenic activities (e.g., harvesting, thinning, clearing, and replanting), carbon is continuously cycled through and among these storage pools, as well as between the forest ecosystem and the atmosphere. For example, as trees grow, carbon is removed from the atmosphere and stored in living tree biomass. As trees age, they continue to accumulate carbon until they reach maturity, at which point carbon storage slows. As trees die and otherwise deposit litter and debris on the forest floor, decay processes release carbon to the atmosphere and also increase soil carbon stocks.

The net change in forest carbon, however, may not be equivalent to the net flux between forests and the atmosphere because timber harvests may not always result in an immediate flux of carbon to the atmosphere. Harvesting in effect transfers carbon from one of the “forest pools” to a “product pool.” Once in a product pool, the carbon is emitted over time as CO₂ if the wood product combusts or decays. The rate of emission varies considerably among different product pools. For example, if timber is harvested for energy use, combustion results in an immediate release of carbon. Conversely, if timber is harvested and subsequently used as lumber in a house, it may be many decades or even centuries before the lumber is allowed to decay and carbon is released to the atmosphere. If wood products are disposed of in landfills, the carbon contained in the wood may be released years or decades later, or may even be stored permanently in the landfill.

This section of the Land-Use Change and Forestry chapter tracks net changes in carbon stocks in five forest carbon pools and two harvested wood pools. The net change in stocks for each pool is estimated, and then the changes in stocks are summed over all pools to estimate total net flux.

An illustration of forest carbon storage pools, and flows between them via emissions, sequestration, and transfers, is presented in Figure 6-1. In this illustration, forest carbon storage pools are represented by boxes, while flows between storage pools, and between storage pools and the atmosphere, are represented by arrows. Note that boxes are not identical with storage pools identified in this chapter. The storage pools identified in this chapter have been arranged to better illustrate the processes that result in transfers of carbon from one pool to another, and that result in emissions to the atmosphere (adapted from Birdsey and Lewis 2001).

Figure 6-1



Approximately 33 percent (747 million acres) of the U.S. land area is forested (Smith et al. 2001). Between 1977 and 1987, forest land declined by approximately 5.9 million acres, and between 1987 and 1997, the area increased by about 9.2 million acres. These changes in forest area represent average annual fluctuations of only about 0.1 percent.

Given the low rate of change in U.S. forest land area, the major influences on the recent net carbon flux from forest land are management activities and the ongoing impacts of previous land-use changes. These activities affect the net flux of carbon by altering the amount of carbon stored in forest ecosystems. For example, intensified management of forests can increase both the rate of growth and the eventual biomass density² of the forest, thereby increasing the uptake of carbon. Harvesting forests removes much of the aboveground carbon, but trees can grow on this area again and sequester carbon. The reversion of cropland to forest land through natural regeneration also will, over decades, result in increased carbon storage in biomass and soils. The net effect of both forest management and land-use change involving forests is captured in these estimates.

In the United States, improved forest management practices, the regeneration of previously cleared forest areas, and timber harvesting and use have resulted in an annual net (i.e., net sequestration) of carbon during the period from 1990 through 2000. Due to improvements in U.S. agricultural productivity, the rate of forest clearing for crop cultivation and pasture slowed in the late 19th century, and by 1920 this practice had all but ceased. As farming expanded in the Midwest and West, large areas of previously cultivated land in the East were taken out of crop production, primarily between 1920 and 1950, and were allowed to revert to forests or were actively reforested. The impacts of these land-use changes are still affecting carbon fluxes from forests in the East. In addition to land-use changes in the early part of this century, carbon fluxes from Eastern forests have been affected by a trend toward managed growth on private land. Collectively, these changes have produced a near doubling of the biomass density in Eastern forests since

the early 1950s. More recently, the 1970s and 1980s saw a resurgence of federally sponsored forest management programs (e.g., the Forestry Incentive Program) and soil conservation programs (e.g., the Conservation Reserve Program), which have focused on tree planting, improving timber management activities, combating soil erosion, and converting marginal cropland to forests. In addition to forest regeneration and management, forest harvests have also affected net carbon fluxes. Because most of the timber that is harvested from U.S. forests is used in wood products and much of the discarded wood products are disposed of by landfilling, rather than incineration, significant quantities of this harvested carbon are transferred to long-term storage pools rather than being released to the atmosphere. The size of these long-term carbon storage pools has also increased over the last century.

Changes in carbon stocks in U.S. forests and harvested wood were estimated to account for an average annual net sequestration of 899 Tg CO₂ Eq. (245 Tg C) over the period 1990 through 2000 (see Table 6-3 and Table 6-4).³ The net sequestration is a reflection of net forest growth and increasing forest area over this period, particularly from 1987 to 1997, as well as net accumulation of carbon in harvested wood pools. The rate of annual sequestration, however, declined by 22 percent between 1990 and 2000. This is due to a greater rate of forest area increase between 1987 and 1997 than between 1997 and 2001. Most of the decline in annual sequestration occurred in the forest soil carbon pool. This is a reflection of modeling assumptions used in this analysis, specifically that soil carbon stocks for each forest type are constant over time, rather than varying by age, whereas biomass carbon stocks are a function of forest type and age class. Therefore, as lands are converted from non-forest to forest, there is a relatively large immediate increase in soil carbon stocks compared to the increase in biomass carbon stocks. The relatively large shifts in annual net sequestration from 1996 to 1997 are the result of calculating average annual forest fluxes from periodic, rather than annual, activity data.

² The term “biomass density” refers to the weight of vegetation per unit area. It is usually measured on a dry-weight basis. Dry biomass is about 50 percent carbon by weight.

³ This average annual net sequestration is based on the entire time series (1990 through 2000), rather than the abbreviated time series presented in Table 6-3 and Table 6-4. Results for the entire time series are presented in Annex N (Methodology for Estimating Net Changes in Forest Carbon Stocks).

Table 6-3: Net CO₂ Flux from U.S. Forests (Tg CO₂ Eq.)

Description	1990	1995	1996	1997	1998	1999	2000
Forest Carbon Stocks	(773.7)	(773.7)	(773.7)	(546.3)	(546.3)	(546.3)	(546.3)
Trees	(469.3)	(469.3)	(469.3)	(447.3)	(447.3)	(447.3)	(447.3)
Understory	(11.0)	(11.0)	(11.0)	(14.7)	(14.7)	(14.7)	(14.7)
Forest Floor	(25.7)	(25.7)	(25.7)	29.3	29.3	29.3	29.3
Down Dead Wood	(55.0)	(55.0)	(55.0)	(58.7)	(58.7)	(58.7)	(58.7)
Forest Soils	(212.7)	(212.7)	(212.7)	(55.0)	(55.0)	(55.0)	(55.0)
Harvested Wood Carbon Stocks	(209.0)	(205.3)	(205.3)	(212.7)	(205.3)	(216.3)	(223.7)
Wood Products	(47.7)	(55.0)	(55.0)	(58.7)	(51.3)	(62.3)	(66.0)
Landfilled Wood	(161.3)	(150.3)	(150.3)	(154.0)	(154.0)	(154.0)	(157.7)
Total	(982.7)	(979.0)	(979.0)	(759.0)	(751.7)	(762.7)	(770.0)

Note: Parentheses indicate net carbon “sequestration” (i.e., accumulation into the carbon pool minus emissions or stock removal from the carbon pool). The sum of the net stock changes in this table (i.e., total) is an estimate of the actual net flux between the total forest carbon pool and the atmosphere. Lightly shaded areas indicate values based on a combination of historical data and projections. Forest values are based on periodic measurements; harvested wood estimates are based on annual surveys. Totals may not sum due to independent rounding.

Table 6-4: Net CO₂ Flux from U.S. Forests (Tg C)

Description	1990	1995	1996	1997	1998	1999	2000
Forest Carbon Stocks	(211)	(211)	(211)	(149)	(149)	(149)	(149)
Trees	(128)	(128)	(128)	(122)	(122)	(122)	(122)
Understory	(3)	(3)	(3)	(4)	(4)	(4)	(4)
Forest Floor	(7)	(7)	(7)	8	8	8	8
Down Dead Wood	(15)	(15)	(15)	(16)	(16)	(16)	(16)
Forest Soils	(58)	(58)	(58)	(15)	(15)	(15)	(15)
Harvested Wood Carbon Stocks	(57)	(56)	(56)	(58)	(56)	(59)	(61)
Wood Products	(13)	(15)	(15)	(16)	(14)	(17)	(18)
Landfilled Wood	(44)	(41)	(41)	(42)	(42)	(42)	(43)
Total	(268)	(267)	(267)	(207)	(205)	(208)	(210)

Note: 1 Tg C = 1 Tg carbon = 1 million metric tons carbon. Parentheses indicate net carbon “sequestration” (i.e., accumulation into the carbon pool minus emissions or harvest from the carbon pool). The sum of the net stock changes in this table (i.e., total) is an estimate of the actual net flux between the total forest carbon pool and the atmosphere. Lightly shaded areas indicate values based on a combination of historical data and projections. Forest values are based on periodic measurements; harvested wood estimates are based on annual surveys. Totals may not sum due to independent rounding.

Methodology

The approach to calculating changes in carbon stocks in forests can generically be described as sampling the forest carbon at one time, sampling the forest carbon a second time at a later date, and then subtracting the two estimates for the net stock change. Historically, the main purpose of the national forest inventory has been to estimate areas, volume of growing stock, and timber products output and utilization factors. Growing stock is simply a classification of timber inventory that includes live trees of commercial species meeting specified standards of quality (Smith et al. 2001). Timber products output refers to the production of industrial roundwood products such as logs and other round

timber generated from harvesting trees, and the production of bark and other residue at processing mills. Utilization factors relate inventory volume to the volume cut or destroyed when producing roundwood (May 1998). Growth, harvests, land-use change, and other estimates of change are derived from repeated surveys. The inventory data are converted to carbon using conversion factors or a model that estimates basic relationships between forest characteristics and carbon pools like forest floor. Historical carbon stock changes are derived from USDA Forest Service, Forest Inventory & Analysis inventory data (Smith et al. 2001, Frayer and Furnival 1999). Projected carbon stock changes are derived from areas, volumes, growth,

land-use changes and other forest characteristics projected in a system of models (see Haynes et al. 2001a) representing the U.S. forest sector, including a model (FORCARB) that estimates carbon for merchantable and non-merchantable tree pools, and other forest carbon pools.

The USDA Forest Service, Forest Inventory & Analysis (FIA) has conducted consistent scientifically designed forest surveys of much of the forest land in the United States since 1952. Historically, these were conducted periodically, state-by-state within a region. One state within a region would be surveyed, and when finished, another state was surveyed. Eventually (every 5-14 years, depending on the state), all states within a region would be surveyed, and then states would be resurveyed. FIA has adopted a new annualized design, so that a portion of each state will be surveyed each year (Gillespie 1999); however, data are not yet available for all states. The annualized survey also includes a plan to measure attributes that are needed to estimate carbon in various pools, such as soil carbon and forest floor carbon. Characteristics that are measured and readily available from some surveys include individual tree diameter and species, and forest type and age of the plot. For more information about forest inventory data and carbon flux, see Birdsey and Heath (2001).

The USDA Forest Service periodically compiles and reports survey data for a specific base year. Available years relevant to CO₂ flux estimates are 1987 and 1997. Live tree carbon and dead tree carbon are estimated from the inventory data using the conversion factors by forest type and region in Smith et al. (in review). Understory carbon is estimated from forest inventory data and equations based on estimates in Birdsey (1992). Forest floor carbon is estimated from the forest inventory data using the equations listed in Smith and Heath (in review). Projections produce estimates of areas and volumes; carbon estimates are produced using this information using procedures similar to those used to produce carbon estimates from forest inventory data. For a detailed description of the modeling system, see Annex N.

In the past, FIA surveyed all productive forest land, which is called timberland, and some reserved forest land and some other forest land.⁴ With the introduction of the annualized design (Gillespie 1999), all forest lands will feature the same type of information. Forest carbon stocks on non-timberland forests were estimated based on average carbon estimates derived from representative timberlands. Reserved forests were assumed to contain the same average carbon densities as timberlands of the same forest type, region, and owner group. These averages were multiplied by the areas in the forest statistics, and then aggregated for a national total. Average carbon stocks were derived for other forest land by using average carbon stocks for timberlands, which were multiplied by 50 percent to simulate the effects of lower productivity.

Estimates of carbon stock changes in wood products and wood discarded in landfills are based on the methods described in Skog and Nicholson (1998). The disposition of harvested wood carbon removed from the forest can be described in four general pools: products in use, discarded wood in landfills, emissions from wood burned for energy, and emissions from decaying wood or wood burned in which energy was not captured. The net carbon stock changes presented here represent the amounts of carbon that are stored (i.e., not released to the atmosphere). Annual historical estimates and projections of detailed production were used to divide consumed roundwood into product, wood mill residue, and pulp mill residue. The carbon decay rates for products and landfills were estimated, and applied to the respective pools. The results were aggregated for national estimates. The production approach to accounting for imports and exports was used. Thus, carbon in exported wood is included using the same disposal rates as in the United States, while carbon in imported wood is not included. Over the period 1990 to 2000, carbon in exported wood accounted for an average of 22 Tg CO₂ Eq. storage per year, with little variation from year to year. For comparison, imports—which are not included in the harvested wood net flux estimates—increased from 26 Tg CO₂ Eq. per year in 1990 to 46 Tg CO₂ Eq. per year in 2000.

⁴ Forest land in the United States includes all land that is at least 10 percent stocked with trees of any size. Timberland is the most productive type of forest land, growing at a rate of 20 cubic feet per acre per year or more. In 1997, there were about 503 million acres of timberlands, which represented 67 percent of all forest lands (Smith and Sheffield 2000). Forest land classified as timberland is unreserved forest land that is producing or is capable of producing crops of industrial wood. The remaining 33 percent of forest land is classified as reserved forest land, which is forest land withdrawn from timber use by statute or regulation, or other forest land, which includes forests on which timber is growing at a rate less than 20 cubic feet per acre per year.

The methodology described above is consistent with the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). The IPCC identifies two approaches to developing estimates of net carbon flux from Land-Use Change and Forestry: 1) using average annual statistics on land use, land-use change, and forest management activities, and applying carbon density and flux rate data to these activity estimates to derive total flux values; or 2) using carbon stock estimates derived from periodic inventories of forest stocks, and measuring net changes in carbon stocks over time. The latter approach was employed because the United States conducts periodic surveys of national forest stocks. In addition, the IPCC identifies two approaches to accounting for carbon emissions from harvested wood: 1) assuming that all of the harvested wood replaces wood products that decay in the inventory year so that the amount of carbon in annual harvests equals annual emissions from harvests; or 2) accounting for the variable rate of decay of harvested wood according to its disposition (e.g., product pool, landfill, combustion). The latter approach was applied for this Inventory using estimates of carbon stored in wood products and landfilled wood.⁵ The use of direct measurements from forest surveys and associated estimates of product and landfilled wood pools is likely to result in more accurate flux estimates than the alternative IPCC methodology.

Data Sources

The estimates of forest carbon stocks used to calculate forest carbon fluxes are based largely on areas, volumes, growth, harvests, and utilization factors derived from the forest inventory data collected by the USDA Forest Service. Compilations of these data for 1987 and 1997 are given in Waddell et al. (1989) and Smith et al. (2001), respectively, with trends discussed in the latter citation. The timber volume data used here include timber volumes on forest land classified as timberland, as well as on some reserved forest land and other forest land. Timber volumes on forest land in Alaska, Hawaii, and the U.S. territories are not sufficiently detailed to be used here. Also, timber volumes on non-forest land (e.g., urban trees, rangeland) are not included. The timber volume data include estimates by tree species, size class, and other categories. The forest inventory

data are augmented or converted to carbon following the methods described in the methodology section. The carbon storage factors applied to these data are described in Annex N. Soil carbon estimates are based on data from the STATSGO database (USDA 1991). Carbon stocks in wood products in use and wood stored in landfills are based on historical data from the USDA Forest Service (USDA 1964, Ulrich 1989, Howard 2001), and historical data as implemented in the framework underlying the NAPAP (Ince 1994) and TAMM/ATLAS (Haynes et al. 2001a, Mills and Kincaid 1992) models. The carbon conversion factors and decay rates for harvested carbon removed from the forest are taken from Skog and Nicholson (1998).

Table 6-5 presents the carbon stock estimates for forest and harvested wood storage pools. Together, the tree and forest soil pools account for over 80 percent of total carbon stocks. Carbon stocks in all pools, except forest floor, increased over time, indicating that, during these periods, all storage pools except forest floor accumulated carbon (e.g., carbon sequestration by trees was greater than carbon removed from the tree pool through respiration, decay, litterfall, and harvest). Figure 6-2 shows 1997 carbon stocks by the regions that were used in the forest carbon analysis.

Uncertainty

There are sampling and measurement errors associated with the forest survey data that underlie the forest carbon estimates. These surveys are based on a statistical sample designed to represent the wide variety of growth conditions present over large territories. Although newer inventories are being conducted annually in every state, much of the data currently used may have been collected over more than one year in a state, and data associated with a particular year may have been collected over several earlier years. Thus, there is uncertainty in the year associated with the forest inventory data. In addition, the forest survey data that are currently available exclude timber stocks on most forest land in Alaska, Hawaii, U.S. territories. The assumptions that were used to calculate carbon stocks in reserved forests and other forests in the coterminous United States also contribute to the uncertainty. Although the potential for uncertainty is large, the sample design for the forest surveys contributes to limiting the error in carbon flux. Re-measured

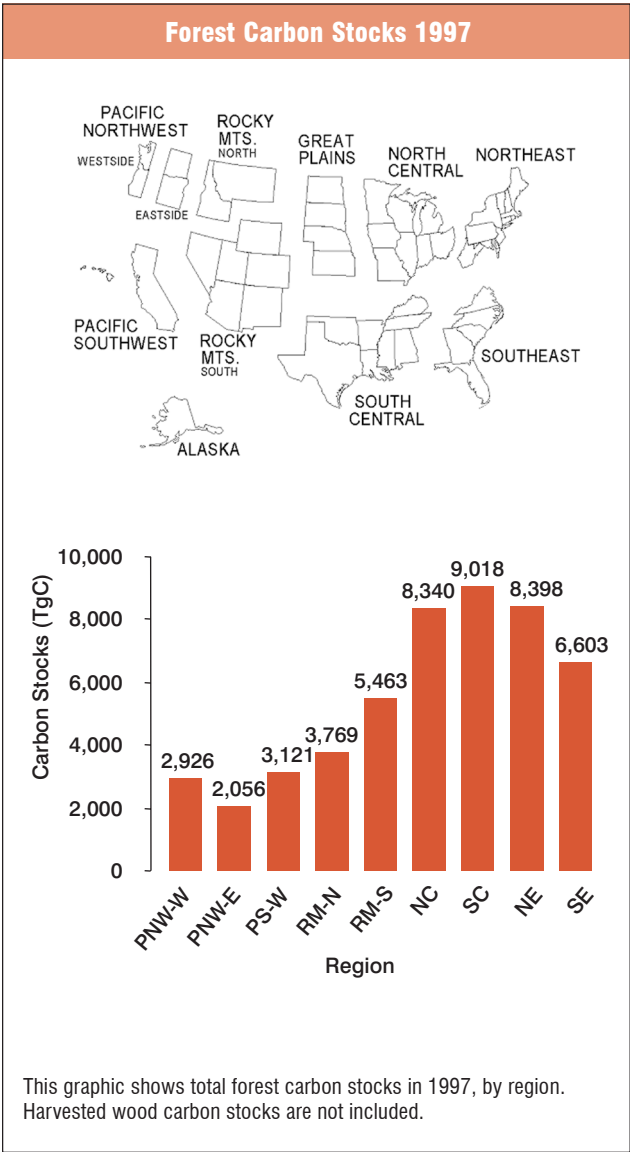
⁵ Again, the product estimates in this study do not account for carbon stored in imported wood products. However, they do include carbon stored in exports, even if the logs are processed in other countries (Heath et al. 1996).

Table 6-5: U.S. Forest Carbon Stock Estimates (Tg C)

Description	1987	1997	2001
Forests	47,594	49,694	50,291
Trees	15,168	16,449	16,937
Understory	448	473	489
Forest Floor	4,240	4,306	4,274
Down Dead Wood	2,058	2,205	2,269
Forest Soils	25,681	26,262	26,322
Harvested Wood	1,920	2,479	2,712
Wood Products	1,185	1,319	1,384
Landfilled Wood	735	1,159	1,328
Total	49,514	52,173	53,003

Note: Forest carbon stocks do not include forest stocks in Alaska, Hawaii, or U.S. territories, or trees on non-forest land (e.g., urban trees); wood product stocks include exports, even if the logs are processed in other countries, and exclude imports. Lightly shaded areas indicate values based on a combination of historical data and projections. All other estimates are based on historical data only. Totals may not sum due to independent rounding. Note that the stock is listed for 2001 because stocks are defined as of January 1 of the listed year.

Figure 6-2



permanent plot estimates are correlated, and greater correlation leads to decreased uncertainties in change estimates. For example, in a study on the uncertainty of the forest carbon budget of private timberlands of the United States, Smith and Heath (2000) estimated that the uncertainty of the flux increased about 3.5 times when the correlation coefficient dropped from 0.95 to 0.5.

Additional sources of uncertainty come from the models used to estimate carbon storage in specific ecosystem components, such as forest floor, understory vegetation, and soil. Extrapolating results of separate ecosystem studies to all forest lands introduces uncertainty through the necessary assumption that the studies adequately describe regional or national averages. These assumptions can potentially introduce the following errors: (1) bias from applying data from studies that inadequately represent average forest conditions, (2) modeling errors (e.g., relying on coefficients or relationships that are not known), and (3) errors in converting estimates from one reporting unit to another (Birdsey and Heath 1995). In particular, the impacts of forest management activities, including harvest, on soil carbon are not well understood. For example, while Johnson and Curtis (2001) found little or no net change in soil carbon following harvest on average across a number of studies, many of the individual studies did exhibit differences. Heath and Smith (2000b) noted that the experimental design in a number of soil studies was such that the usefulness of the studies may be limited in determining harvesting effects on soil carbon. Soil carbon impact estimates need to be precise; even small

changes in soil carbon may sum to large differences over large areas. This analysis assumes that soil carbon density for each forest type stays constant over time. In the future, land-use effects will be incorporated into the soil carbon density estimates.

Recent studies have looked at quantifying the amount of uncertainty in national-level carbon budgets based on the methods adopted here. Smith and Heath (2000) and Heath and Smith (2000a) report on an uncertainty analysis they conducted on carbon sequestration in private timberlands. These studies are not strictly comparable to the estimates in this chapter because they used an older version of the FORCARB model, which was based on older data and produced decadal estimates. However, the magnitudes of the uncertainties should be instructive. Their results indicate that the carbon flux of private timberlands, not including harvested wood, was approximately the average carbon flux (271 Tg CO₂ Eq. per year) ±15 percent at the 80 percent confidence level for the period 1990 through 1999. The flux estimate included the tree, soil, understory vegetation, and forest floor components only. The uncertainty in the carbon inventory of private timberlands for 2000 was approximately 5 percent at the 80 percent confidence level. These estimates did not include all uncertainties, such as the ones associated with public timberlands, and reserved and other forest land, but they did include many of the types of uncertainties listed previously. It is expected that the uncertainty should be greater for all forest lands.

Changes in Carbon Stocks in Urban Trees

Urban forests constitute a significant portion of the total U.S. tree canopy cover (Dwyer et al. 2000). It was estimated that urban areas (cities, towns, and villages), which cover 3.5 percent of the continental United States, contained about 3.8 billion trees. With an average tree canopy cover of 27.1 percent, urban areas accounted for approximately 2.8 percent of total tree cover in the continental United States.

Trees in urban areas of the continental United States were estimated by Nowak and Crane (2001) to account for an average annual net sequestration of 59 Tg CO₂ Eq. (16 Tg C). This estimate is representative of the period from 1990 through 2000, as it is based on data collected during that decade. Annual estimates of CO₂ flux have not been developed (see Table 6-6).

Methodology

The methodology used by Nowak and Crane (2001) is based on average annual estimates of urban tree growth and decomposition, which were derived from field measurements and data from the scientific literature, urban area estimates from U.S. Census data, and urban tree cover estimates from remote sensing data. This approach is consistent with, but more robust than, the default IPCC methodology in the *Revised 1996 IPCC Guidelines* (IPCC/ UNEP/OECD/IEA 1997).⁶

Nowak and Crane (2001) developed estimates of annual gross carbon sequestration from tree growth and annual gross carbon emissions from decomposition for ten U.S. cities: Atlanta, GA; Baltimore, MD; Boston, MA; Chicago, IL; Jersey City, NJ; New York, NY; Oakland, CA; Philadelphia, PA; Sacramento, CA; and Syracuse, NY. The gross carbon sequestration estimates were derived from field data that were collected in these ten cities during the period from 1990 through 2000, including tree measurements of stem diameter, tree height, crown height, and crown width, and information on location, species, and canopy condition. The field data were converted to annual gross carbon sequestration rates for each species (or genus), diameter

Table 6-6: Net CO₂ Flux From Urban Trees (Tg CO₂ Eq.)

Year	Tg CO ₂ Eq.
1990	(58.7)
1995	(58.7)
1996	(58.7)
1997	(58.7)
1998	(58.7)
1999	(58.7)
2000	(58.7)

Note: Parentheses indicate net sequestration.

⁶ It is more robust in that both growth and decomposition are accounted for, and data from individual trees are scaled up to state and then national estimates based on data on urban area and urban tree canopy cover.

Table 6-7: Carbon Storage (Metric Tons C), Carbon Sequestration (Metric Tons C/yr), and Tree Cover (%) for Ten U.S. Cities

City	Carbon Storage	Gross Sequestration	Net Sequestration	Tree Cover
New York, NY	1,225,200	38,400	20,800	20.9%
Atlanta, GA	1,220,200	42,100	32,200	36.7%
Sacramento, CA	1,107,300	20,200	NA	13.0%
Chicago, IL	854,800	40,100	NA	11.0%
Baltimore, MD	528,700	14,800	10,800	25.2%
Philadelphia, PA	481,000	14,600	10,700	15.7%
Boston, MA	289,800	9,500	6,900	22.3%
Syracuse, NY	148,300	4,700	3,500	24.4%
Oakland, CA	145,800	NA	NA	21.0%
Jersey City, NJ	19,300	800	600	11.5%
NA (Not Available)				

class, and land-use condition (forested, park-like, and open growth) by applying allometric equations, a root-to-shoot ratio, moisture contents, a carbon content of 50 percent (dry weight basis), an adjustment factor to account for smaller aboveground biomass volumes (given a particular diameter) in urban conditions compared to forests, an adjustment factor to account for tree condition (fair to excellent, poor, critical, dying, or dead), and annual diameter and height growth rates. The annual gross carbon sequestration rates for each species (or genus), diameter class, and land-use condition were then scaled up to city estimates using tree population information (see Table 6-7).

The annual gross carbon emission estimates were derived by applying to carbon stock estimates, which were derived as an intermediate step in the gross sequestration calculations, estimates of annual mortality by tree diameter and condition class, assumptions about whether dead trees would be removed from the site—since removed trees were assumed to decay faster than those left on the site—and assumed decomposition rates for dead trees left standing and dead trees that are removed. The annual gross carbon emission rates for each species (or genus), diameter class, and condition class were then scaled up to city estimates using tree population information.

Annual net carbon sequestration estimates were derived for each of the ten cities by subtracting by the annual gross emission estimates from the annual gross sequestration estimates (see Table 6-7).

National annual net carbon sequestration by urban trees was estimated from the city estimates of gross and net sequestration, and urban area and urban tree cover data for the contiguous United States. Note that the urban areas are based on U.S. Census data, which define “urban” as having a population greater than 2,500. Therefore, urban encompasses most cities, towns, and villages (i.e., it includes both urban and suburban areas). The gross and net carbon sequestration values for each city were divided by each city’s area of tree cover to determine the average annual sequestration rates per unit of tree area for each city (see Table 6-8). The median value for gross sequestration (0.30 kg C/m²-year) was then multiplied by an estimate of national urban tree cover area (76,151 km²) to estimate national annual gross sequestration. To estimate national annual net sequestration, the estimate of national annual gross sequestration was multiplied by the average of the ratios of net to gross sequestration for those cities that had both estimates (0.70).

Table 6-8: Annual Sequestration per Area of Tree Cover (kg C/m² cover-year)

City	Gross	Net
New York, NY	0.23	0.12
Atlanta, GA	0.34	0.26
Sacramento, CA	0.66	NA
Chicago, IL	0.61	NA
Baltimore, MD	0.28	0.20
Philadelphia, PA	0.27	0.20
Boston, MA	0.30	0.22
Syracuse, NY	0.30	0.22
Oakland, CA	NA	NA
Jersey City, NJ	0.18	0.13
NA (Not Available)		

Data Sources

The field data from the 10 cities, some of which are unpublished, are described in Nowak and Crane (2001) and references cited therein. The allometric equations were taken from the scientific literature (see Nowak 1994, Nowak et al. in press), and the adjustments to account for smaller volumes in urban conditions were based on information in Nowak (1994). A root-to-shoot ratio of 0.26 was taken from Cairns et al. (1997), and species- or genus-specific moisture contents were taken from various literature sources (see Nowak 1994). Adjustment factors to account for tree condition were based on expert judgement of the authors (Nowak and Crane 2001). Tree growth rates were also taken from existing literature. Average diameter growth was based on the following sources: estimates for trees in forest stands came from Smith and Shifley (1984); estimates for trees on land uses with a park-like structure came from deVries (1987); and estimates for more open-grown trees came from Nowak (1994). Formulas from Fleming (1988) formed the basis for average height growth calculations. Estimates of annual mortality rates by diameter class and condition class were derived from a study of street-tree mortality (Nowak 1986). Assumptions about whether dead trees would be removed from the site and assumed decomposition rates were based on percent crown dieback (Nowak and Crane 2001). Urban tree cover area estimates for each of the 10 cities and the contiguous United States were obtained from Dwyer et al. (2000) and Nowak et al. (2001).

Uncertainty

The estimates are based on limited field data collected in ten U.S. cities, and the uncertainty in these estimates increases as they are scaled up to the national level. There is also uncertainty associated with the biomass equations, conversion factors, and decomposition assumptions used to calculate carbon sequestration and emission estimates (Nowak et al. in press), as well as with the tree cover area estimates for urban areas, as these are based on interpretation of Advanced Very High Resolution Radiometer (AVHRR) data. In addition, these results do not include changes in soil carbon stocks, and there may be some overlap between the urban tree carbon estimates and the forest tree carbon estimates. However, both the omission of urban soil carbon flux, and the potential overlap with forest carbon, are believed to be relatively minor (Nowak 2002).

Changes in Agricultural Soil Carbon Stocks

The amount of organic carbon contained in soils depends on the balance between inputs of organic material (e.g., decayed plant matter, roots, and organic amendments such as manure and crop residues) and loss of carbon through decomposition. The quantity and quality of organic matter inputs, and their rate of decomposition, are determined by the combined interaction of climate, soil properties, and land use. Agricultural practices such as clearing, drainage, tillage, planting, grazing, crop residue management, fertilization, and flooding, can modify both organic matter inputs and decomposition, and thereby result in a net flux of carbon to or from soils. In addition, the application of carbonate minerals to soils through liming operations results in emissions of CO₂. The IPCC methodology for estimation of net CO₂ flux from agricultural soils (IPCC/UNEP/OECD/IEA 1997) is divided into three categories of land-use/land-management activities: 1) agricultural land-use and land-management activities on mineral soils; 2) agricultural land-use and land-management activities on organic soils; and 3) liming of soils. Mineral soils and organic soils are treated separately because each responds differently to land-use practices.

Mineral soils contain comparatively low amounts of organic matter, much of which is concentrated near the soil surface. Typical well-drained mineral surface soils contain from 1 to 6 percent organic matter (by weight); mineral subsoils contain even lower amounts of organic matter (Brady and Weil 1999). When mineral soils undergo conversion from their native state to agricultural use, as much as half of the soil organic carbon can be lost to the atmosphere. The rate and ultimate magnitude of carbon loss will depend on native vegetation, conversion method and subsequent management practices, climate, and soil type. In the tropics, 40 to 60 percent of the carbon loss generally occurs within the first 10 years following conversion; after that, carbon stocks continue to decline but at a much slower rate. In temperate regions, carbon loss can continue for several decades. Eventually, the soil will reach a new equilibrium that reflects a balance between carbon accumulation from plant biomass and carbon loss through oxidation. Any changes in land-use or management practices that result in increased organic inputs or decreased oxidation of organic matter (e.g.,

Table 6-9: Net CO₂ Flux From Agricultural Soils (Tg CO₂ Eq.)

Description	1990		1995	1996	1997	1998	1999	2000
Mineral Soils	(69.3)		(92.0)	(92.0)	(92.0)	(99.7)	(99.7)	(99.7)
Organic Soils	22.5		22.9	22.9	22.9	22.9	22.9	22.9
Liming of Soils	9.5		8.9	8.9	8.7	9.6	9.1	9.4
Total	(37.3)		(60.2)	(60.2)	(60.4)	(67.2)	(67.7)	(67.4)

Note: Parentheses indicate net sequestration. Lightly shaded areas indicate values based on a combination of historical data and projections. All other values are based on historical data only.

improved crop rotations, cover crops, application of organic amendments and manure, and reduction or elimination of tillage) will result in a net accumulation of soil organic carbon until a new equilibrium is achieved.

Organic soils, which are also referred to as histosols, include all soils with more than 20 to 30 percent organic matter by weight, depending on clay content (Brady and Weil 1999). The organic matter layer of these soils is also typically extremely deep. Organic soils form under water-logged conditions, in which decomposition of plant residues is retarded. When organic soils are cultivated, they are first drained which, together with tilling or mixing of the soil, aerates the soil, and thereby accelerates the rate of decomposition and CO₂ generation. Because of the depth and richness of the organic layers, carbon loss from cultivated organic soils can continue over long periods of time. When organic soils are disturbed, through cultivation and/or drainage, the rate at which organic matter decomposes, and therefore the rate at which CO₂ emissions are generated, is determined primarily by climate, the composition (i.e., decomposability) of the organic matter, and the specific land-use practices undertaken. The use of organic soils for annual crops results in greater carbon loss than conversion to pasture or forests, due to deeper drainage and more intensive management practices (Armentano and Verhoeven 1990, as cited in IPCC/UNEP/OECD/IEA 1997).

Lime in the form of crushed limestone (CaCO₃) and dolomite (CaMg(CO₃)₂) is commonly added to agricultural soils to ameliorate acidification. When these compounds come in contact with acid soils, they degrade, thereby generating CO₂. The rate of degradation is determined by soil conditions and the type of mineral applied; it can take several years for applied limestone and dolomite to degrade completely.

Of the three activities, use and management of mineral soils was the most important component of total flux during the 1990 through 2000 period (see Table 6-9). Carbon sequestration in mineral soils in 2000 was estimated at about 100 Tg CO₂ Eq. (27 Tg C), while emissions from organic soils were estimated at 23 Tg CO₂ Eq. (6 Tg C) and emissions from liming were estimated at 9 Tg CO₂ Eq. (3 Tg C). Together, the three activities accounted for net sequestration of about 67 Tg CO₂ Eq. (18 Tg C) in 2000. Total annual net CO₂ flux was negative (i.e., net sequestration) each year over the 1990 to 2000 period. Between 1990 and 2000, total net carbon sequestration in agricultural soils increased by about 80 percent. The increase is largely due to additional acreage of annual cropland converted to permanent pastures and hay production, a reduction in the frequency of summer-fallow use in semi-arid areas and some increase in the adoption of conservation tillage (i.e., reduced and no-till) practices. The relatively large shifts in annual net sequestration from 1990 to 1995, and from 1997 to 1998 are the result of calculating average annual mineral and organic soil fluxes from periodic, rather than annual, activity data. The results for mineral and organic soils are displayed by region in Figure 6-3, Figure 6-4, Figure 6-5, and Figure 6-6.

The flux estimates presented here are restricted to CO₂ fluxes associated with the use and management of agricultural soils. Agricultural soils are also important sources of other greenhouse gases, particularly nitrous oxide (N₂O) from application of fertilizers, manure, and crop residues and from cultivation of legumes, as well as methane (CH₄) from flooded rice cultivation. These emissions are accounted for in the Agriculture chapter.⁷ It should be noted that other land-use and land-use change activities result in fluxes of non-CO₂ greenhouse gases to and from soils that

⁷ Nitrous oxide emissions from agricultural soils and methane emissions from rice fields are addressed under the Agricultural Soil Management and Rice Cultivation sections, respectively, of the Agriculture chapter.

Figure 6-3

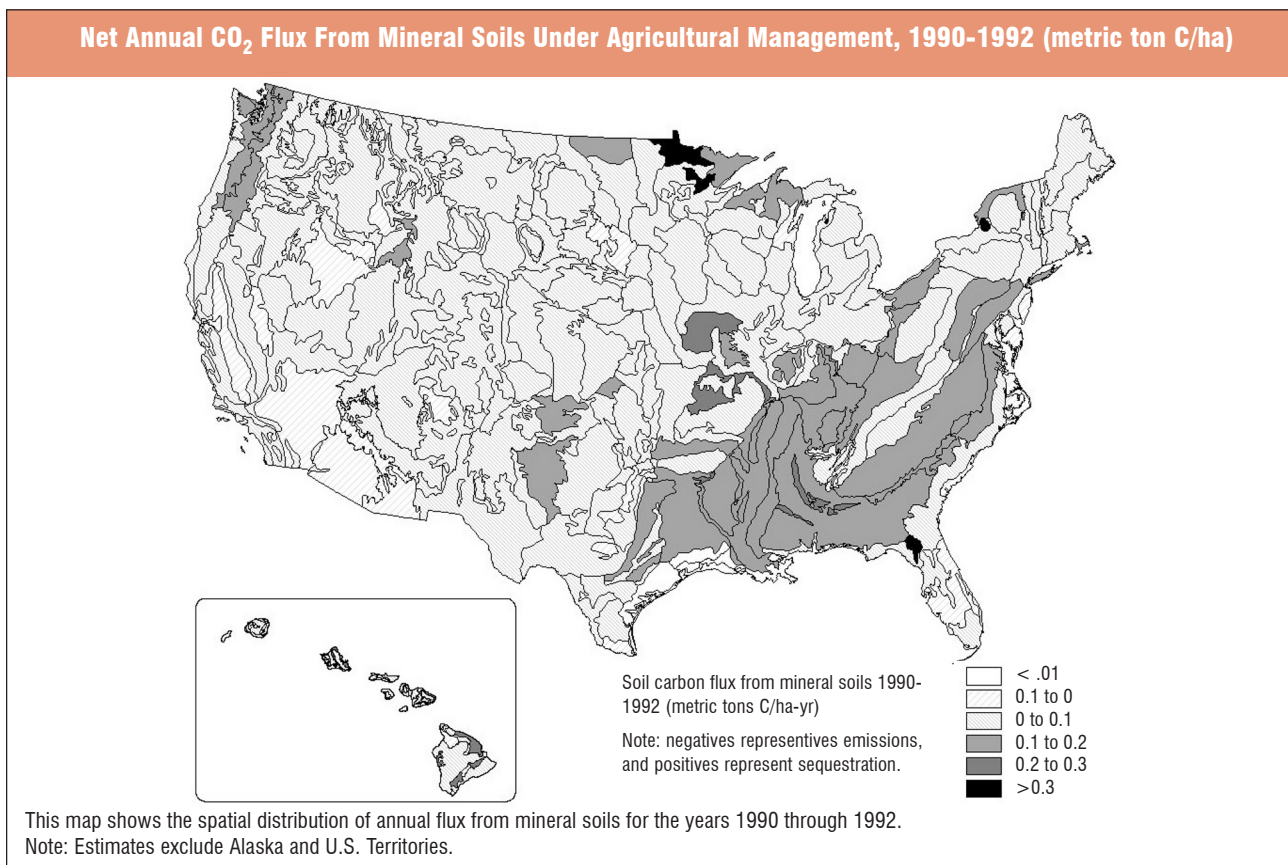


Figure 6-4

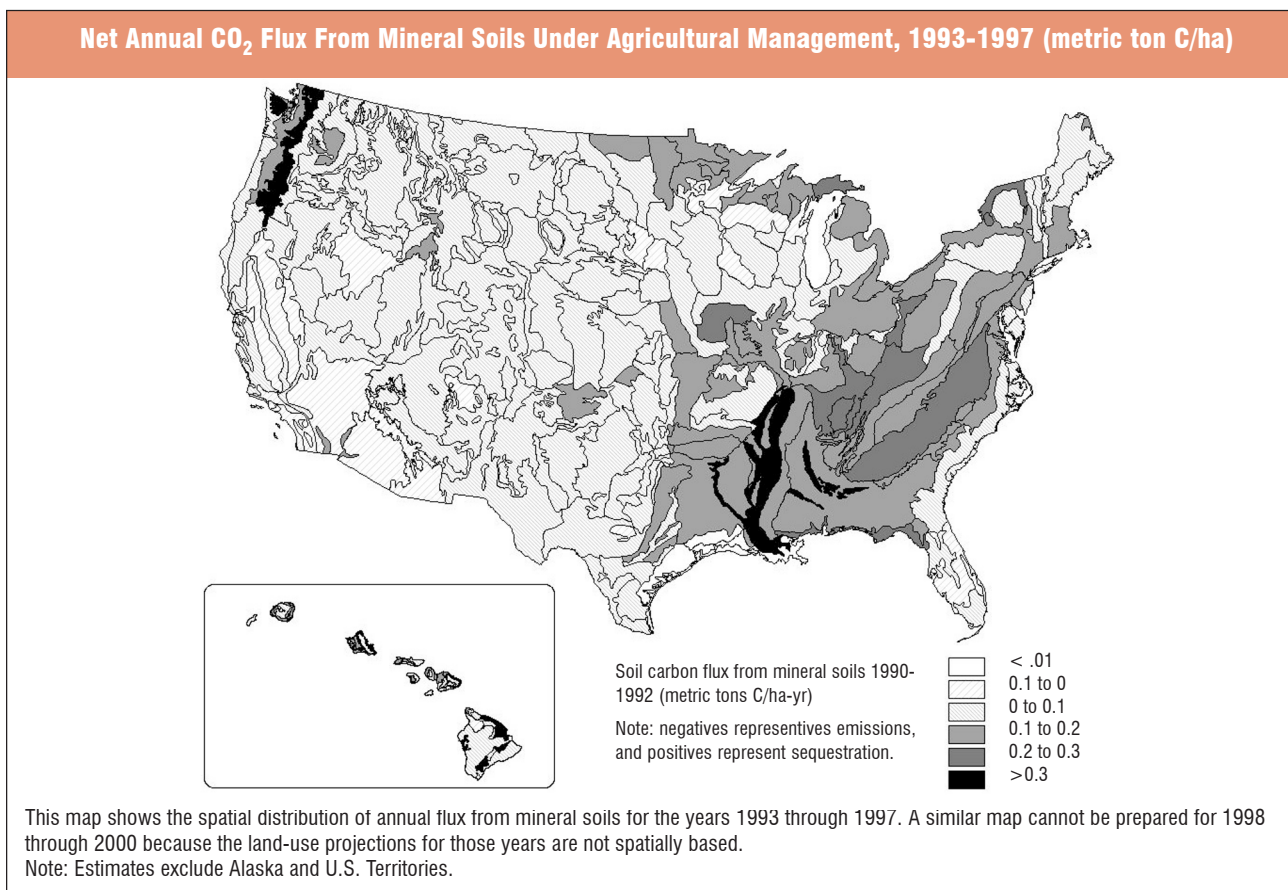
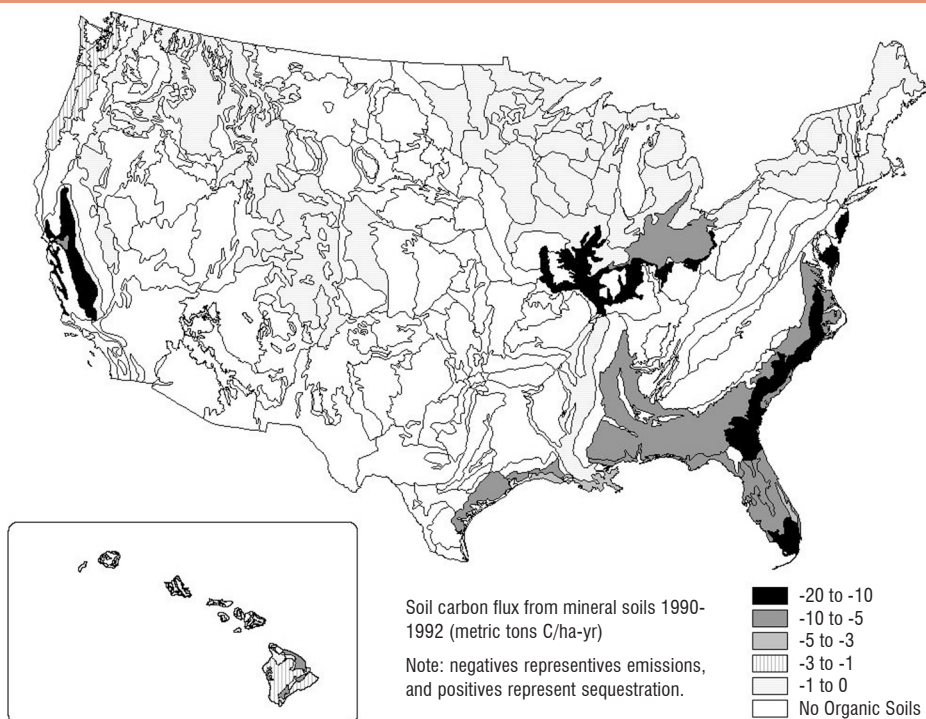


Figure 6-5

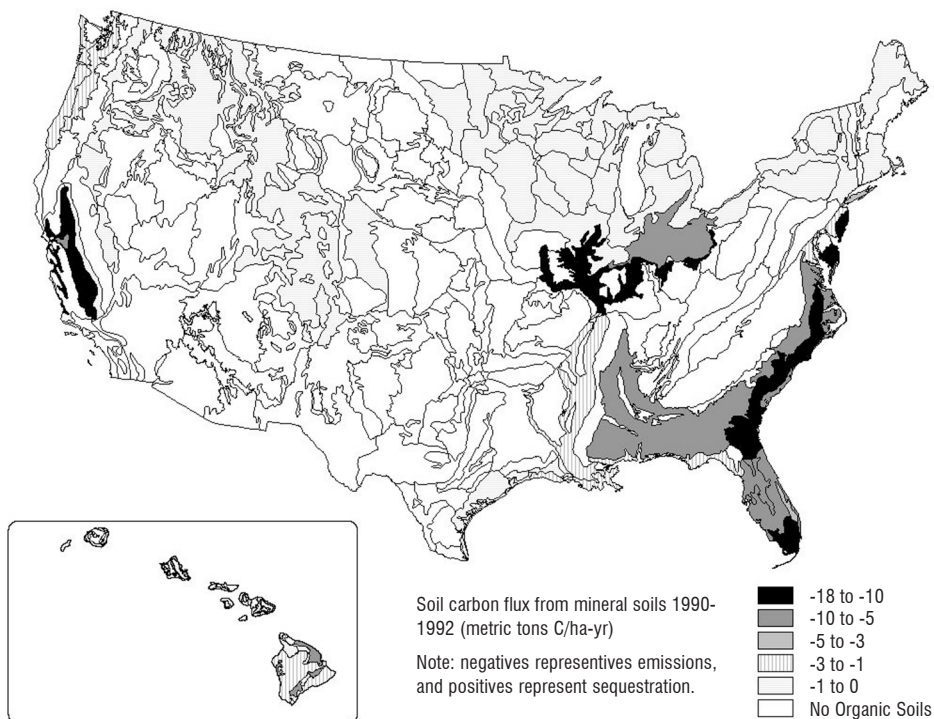
Net Annual CO₂ Flux From Organic Soils Under Agricultural Management, 1990-1992 (metric ton C/ha)



This map shows the spatial distribution of annual flux from organic soils for the years 1993 through 2000.

Figure 6-6

Net Annual CO₂ Flux From Organic Soils Under Agricultural Management, 1993-2000 (metric ton C/ha)



This map shows the spatial distribution of annual flux from organic soils for the years 1993 through 2000.

are not currently accounted for. These include emissions of CH₄ and N₂O from managed forest soils (above what would occur if the forest soils were undisturbed), as well as CH₄ emissions from artificially flooded lands, resulting from activities such as dam construction. Aerobic (i.e., non-flooded) soils are a sink for CH₄, so soil drainage can result in soils changing from a CH₄ source to a CH₄ sink, but if the drained soils are used for agriculture, fertilization and tillage disturbance can reduce the ability of soils to oxidize CH₄. The non-CO₂ emissions and sinks from these other land use and land-use change activities were not assessed due to scientific uncertainties about the greenhouse gas fluxes that result from these activities.

Methodology and Data Sources

The methodologies used to calculate CO₂ emissions from use and management of mineral and organic soils and from liming follow the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997), except where noted below.

The estimates of annual net CO₂ flux from mineral soils were based on application of the *Revised 1996 IPCC Guidelines* as described by Eve et al. (2001). Total mineral soil carbon stock estimates for 1982, 1992, and 1997 were developed for the conterminous United States and Hawaii by applying the default IPCC carbon stock and carbon adjustment factors (with one exception), to cropland and grazing land area estimates, classified by climate, soil type, and management regime. The exception is the base factor for lands set aside for less than 20 years. The IPCC default value is 0.8, but recent research (e.g., Paustian et al. 2001, Follett et al. 2001, Huggins et al. 1998, and Gebhart et al. 1994) indicates that 0.9 is a more accurate factor for the United States. Therefore, 0.9 was used instead of 0.8 for the base factor for grassland set aside through the Conservation Reserve Program. Data on land-use and land-management changes over time were aggregated by Major Land Resource Areas (MLRAs; NRCS 1981), which represent geographic units with relatively similar soils, climate, water resources, and land uses. MLRAs were

Table 6-10: Mineral Soil Areas by Land-Use Category and IPCC Climatic Region (million hectares)^a

Category	1982	1992	1997
Cropland (No fallow)	117.6	107.6	109.2
Cold Temperate, Dry	2.5	2.4	2.8
Cold Temperate, Moist	44.9	42.9	45.6
Warm Temperate, Dry	12.4	10.7	10.6
Warm Temperate, Moist	53.9	47.9	46.7
Sub-Tropical, Dry	1.3	1.2	1.1
Sub-Tropical, Moist	2.6	2.6	2.4
Cropland (Fallow)	27.4	24.6	20.0
Cold Temperate, Dry	9.4	7.6	7.0
Cold Temperate, Moist	9.0	8.0	5.2
Warm Temperate, Dry	4.8	4.2	4.0
Warm Temperate, Moist	4.0	4.7	3.5
Sub-Tropical, Dry	0.0	0.0	0.0
Sub-Tropical, Moist	0.2	0.2	0.2
Hay	19.6	21.1	22.0
Cold Temperate, Dry	2.1	2.2	2.4
Cold Temperate, Moist	10.7	11.0	10.9
Warm Temperate, Dry	0.9	1.2	1.2
Warm Temperate, Moist	5.7	6.5	7.5
Sub-Tropical, Dry	0.1	0.1	0.1
Sub-Tropical, Moist	0.0	0.0	0.0
Grazing Land	214.1	215.6	212.5
Cold Temperate, Dry	38.3	38.3	38.2
Cold Temperate, Moist	48.3	47.4	46.5
Warm Temperate, Dry	51.0	51.5	51.2
Warm Temperate, Moist	64.2	66.2	64.6
Sub-Tropical, Dry	6.8	6.9	6.9
Sub-Tropical, Moist	5.3	5.3	5.0

Category	1982	1992	1997
CRP^b	0.0	13.7	13.2
Cold Temperate, Dry	0.0	2.4	2.3
Cold Temperate, Moist	0.0	4.4	4.1
Warm Temperate, Dry	0.0	2.5	2.4
Warm Temperate, Moist	0.0	4.4	4.3
Sub-Tropical, Dry	0.0	0.1	0.1
Sub-Tropical, Moist	0.0	0.0	0.0
Non-Agricultural^c	6.3	2.5	8.1
Cold Temperate, Dry	0.8	0.2	0.4
Cold Temperate, Moist	1.2	0.5	1.9
Warm Temperate, Dry	1.0	0.3	0.8
Warm Temperate, Moist	3.1	1.4	4.4
Sub-Tropical, Dry	0.0	0.0	0.1
Sub-Tropical, Moist	0.1	0.1	0.6
Total	385.0	385.0	385.0
Cold Temperate, Dry	53.1	53.1	53.1
Cold Temperate, Moist	114.2	114.2	114.2
Warm Temperate, Dry	70.2	70.2	70.2
Warm Temperate, Moist	131.0	131.0	131.0
Sub-Tropical, Dry	8.3	8.3	8.3
Sub-Tropical, Moist	8.3	8.3	8.3

Note: Totals may not sum due to independent rounding.
^a Based on analysis of the 1997 *National Resources Inventory* data (NRCS 2000). Includes all conterminous U.S. land categorized as agricultural in 1992 or 1997.
^b CRP (Conservation Reserve Program)
^c Non-agricultural lands are included when they are either cropland or grazing land during 1992 or 1997.

Table 6-11: Mineral Soil Areas by Land-Use Category and IPCC Mineral Soil Category (thousand hectares)^a

Category	1982	1992	1997
Cropland (Fallow)	27,338	24,627	20,024
High Clay Activity Mineral Soils	24,026	21,153	17,422
Low Clay Activity Mineral Soils	1,516	1,370	1,160
Sandy Soils	635	613	416
Volcanic Soils	11	11	12
Aquic Soils	1,149	1,481	1,016
Cropland (No fallow)	117,825	107,563	109,194
High Clay Activity Mineral Soils	72,043	66,093	68,437
Low Clay Activity Mineral Soils	14,151	11,960	11,292
Sandy Soils	9,198	7,919	7,645
Volcanic Soils	163	137	130
Aquic Soils	22,270	21,454	21,690
CRP^b	0	13,745	13,209
High Clay Activity Mineral Soils	0	10,087	9,671
Low Clay Activity Mineral Soils	0	1,555	1,491
Sandy Soils	0	1,259	1,219
Volcanic Soils	0	18	17
Aquic Soils	0	826	811
Grazing Land	213,840	215,585	212,540
High Clay Activity Mineral Soils	136,731	136,669	135,273
Low Clay Activity Mineral Soils	41,876	43,445	42,665
Sandy Soils	24,885	25,126	24,667
Volcanic Soils	362	381	365
Aquic Soils	9,986	9,965	9,570
Hay	19,616	21,056	22,001
High Clay Activity Mineral Soils	13,563	14,227	14,719
Low Clay Activity Mineral Soils	2,745	3,225	3,621
Sandy Soils	1,132	1,318	1,369
Volcanic Soils	228	240	231
Aquic Soils	1,948	2,047	2,061
Non-Agricultural^c	6,426	2,468	8,075
High Clay Activity Mineral Soils	2,782	918	3,624
Low Clay Activity Mineral Soils	2,280	1,012	2,337
Sandy Soils	651	266	1,184
Volcanic Soil	48	26	57
Aquic Soils	665	246	873
Total	385,044	385,044	385,044
High Clay Activity Mineral Soils	249,146	249,146	249,146
Low Clay Activity Mineral Soils	62,567	62,567	62,567
Sandy Soils	36,500	36,500	36,500
Volcanic Soils	811	811	811
Aquic Soils	36,019	36,019	36,019

Note: Totals may not sum due to independent rounding.

^a Based on analysis of the 1997 *National Resources Inventory* data (NRCS 2000). Includes all conterminous U.S. land categorized as agricultural in 1992 or 1997.

^b CRP (Conservation Reserve Program)

^c Non-agricultural land are included when they are either cropland or grazing land during 1992 or 1997.

classified by IPCC climate categories using the climate mapping program in Daly et al. (1994). For each MLRA, area estimates for each combination of soil type and land-use/land-management combination were derived for 1982, 1992, and 1997 using data obtained from the 1997 *National Resources Inventory* (NRI; NRCS 2000). Mineral soil areas by broad land-use category and IPCC climatic region, and by broad land-use category and IPCC mineral soil category, are shown in Table 6-10 and Table 6-11, respectively. Estimates of tillage practices for each cropping system were derived from data collected by the Conservation Technology Information Center (CTIC 1998), as adjusted by Towery (2001) (see Table 6-12).

The carbon flux estimate for 1990 is based on the change in stocks between 1982 and 1992, and the carbon flux estimate for 1995 through 1997 is based on the change in stocks between 1982 and 1997. The IPCC base, tillage, and input factors were adjusted to account for use of a ten-year and a fifteen-year accounting period, rather than the 20-year period used in the *Revised 1996 IPCC Guidelines*. The carbon flux estimates for 1998 through 2000 were based on a projection of 1997 land use and management to 2008 (USDA 2000b).

The estimates of annual CO₂ emissions from organic soils were also based on the *Revised 1996 IPCC Guidelines* as described by Eve et al. (2001). The IPCC methodology for organic soils utilizes annual CO₂ emission factors, rather than a stock change approach. Following the IPCC methodology, only organic soils under intense management were included, and the default IPCC rates of carbon loss were applied to the total 1992 and 1997 areas for the climate/land-use categories defined in the IPCC Guidelines (see Table 6-13).⁸ The area estimates were derived from the same climatic, soil, and land-use/land-management databases that were used in the mineral soil calculations (Daly et al. 1994, USDA 2000a). The annual flux estimated for 1992 is applied to 1990, and the annual flux estimated for 1997 is applied to 1995 through 2000.

⁸ The default IPCC emission factors for tropical regions was applied to the sub-tropical areas.

Table 6-12: Tillage Percentages by Management Category and IPCC Climatic Zone^a

Climatic Region/Cropping System	1982			1992			1997		
	No Till ^b	Reduced Till ^c	Conv. Till ^d	No Till ^b	Reduced Till ^c	Conv. Till ^d	No Till ^b	Reduced Till ^c	Conv. Till ^d
Sub-Tropical, Dry									
Continuous Cropping Rotations ^e	0	3	97	0	4	96	0	15	85
Rotations with Fallow ^f	0	0	100	0	2	98	0	5	95
Low Residue Agriculture ^g	0	3	97	0	4	96	0	10	90
Sub-Tropical, Moist									
Continuous Cropping Rotations ^e	0	0	100	0	20	80	1	10	89
Rotations with Fallow ^f	0	0	100	0	10	90	1	10	89
Low Residue Agriculture ^g	0	3	97	0	4	96	0	5	95
Warm Temperate, Dry									
Continuous Cropping Rotations ^e	0	0	100	0	10	90	1	15	84
Rotations with Fallow ^f	0	3	97	0	15	85	2	20	78
Low Residue Agriculture ^g	0	3	97	0	1	99	0	0	100
Warm Temperate, Moist									
Continuous Cropping Rotations ^e	0	6	94	10	30	60	12	28	60
Rotations with Fallow ^f	0	6	94	5	30	65	8	27	65
Low Residue Agriculture ^g	0	9	91	1	10	89	2	13	85
Cold Temperate, Dry									
Continuous Cropping Rotations ^e	0	3	97	2	25	73	8	12	80
Rotations with Fallow ^f	0	6	94	4	25	71	12	13	75
Low Residue Agriculture ^g	0	0	100	1	2	97	2	6	92
Cold Temperate, Moist									
Continuous Cropping Rotations ^e	0	11	89	5	30	65	3	17	80
Rotations with Fallow ^f	0	11	89	5	30	65	3	27	70
Low Residue Agriculture ^g	0	0	100	1	2	97	1	7	92

^a Based on annual survey conducted by Conservation Technology Information Center (CTIC), with modifications for long-term adoption of no-till agriculture (Towery 2001).

^b No-till includes CTIC survey data designated as no-tillage.

^c Conventional till includes CTIC survey data designated as intensive tillage and conventional tillage.

^d Reduced-till includes CTIC survey data designated as ridge tillage, mulch tillage, and reduced tillage.

^e Includes medium and high input rotations. CTIC survey data for corn, soybeans, and sorghum were used in this category.

^f Includes rotations with fallow. CTIC survey data on fallow and small grain cropland were used in this category.

^g Includes low input rotations (low residue crops and vegetables in rotation). CTIC survey data on cotton were used in this category.

Carbon dioxide emissions from degradation of limestone and dolomite applied to agricultural soils were calculated by multiplying the annual amounts of limestone and dolomite applied (see Table 6-14) by CO₂ emission factors (0.120 metric ton C/metric ton limestone, 0.130 metric ton C/metric ton dolomite).⁹ These emission factors are based on the assumption that all of the carbon in these materials evolves as CO₂ in the same year in which the minerals are applied. The annual application rates of limestone and dolomite were derived from estimates and industry statistics provided in the *Minerals Yearbook* and *Mineral Industry Surveys* (Tepordei 1993, 1994, 1995,

1996, 1997, 1998, 1999, 2000, 2001; USGS 2001). To develop these data, USGS (U.S. Bureau of Mines prior to 1997) obtained production and use information by surveying crushed stone manufacturers. Because some manufacturers were reluctant to provide information, the estimates of total crushed limestone and dolomite production and use were divided into three components: 1) production by end-use, as reported by manufacturers (i.e., “specified” production); 2) production reported by manufacturers without end-uses specified (i.e., “unspecified” production); and 3) estimated additional production by manufacturers who did not respond to the survey (i.e., “estimated” production).

⁹ Note: the default emission factor for dolomite provided in the Workbook volume of the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997) is incorrect. The value provided is 0.122 metric ton carbon/metric ton of dolomite; the correct value is 0.130 metric ton carbon/metric ton of dolomite.

Table 6-13: Organic Soil Areas by IPCC Land-Use Category and Climatic Region (thousand hectares)^a

Climatic Region/ Land-Use Category	1982	1992	1997
Cold Temperate, Dry	4	4	4
Non-Agricultural ^b	3	3	3
Pasture/Forest	1	1	1
Cropland	0	0	0
Cold Temperate, Moist	757	757	757
Non-Agricultural ^b	79	53	52
Pasture/Forest	368	419	411
Cropland	310	285	294
Sub-Tropical, Dry	2	2	2
Non-Agricultural ^b	2	2	2
Pasture/Forest	0	0	0
Cropland	0	0	0
Sub-Tropical, Moist	391	391	391
Non-Agricultural ^b	143	131	117
Pasture/Forest	63	66	77
Cropland	185	194	196
Warm Temperate, Dry	47	47	47
Non-Agricultural ^b	1	<1	1
Pasture/Forest	2	1	2
Cropland	44	45	44
Warm Temperate, Moist	140	140	140
Non-Agricultural ^b	13	3	2
Pasture/Forest	34	39	38
Cropland	93	98	101
Total	1,341	1,341	1,341
Non-Agricultural ^b	240	193	176
Pasture/Forest	469	526	530
Cropland	633	623	635

Note: Totals may not sum due to independent rounding.

^a Based on analysis of the 1997 *National Resources Inventory* data (NRCS 2000). Includes all conterminous U.S. land categorized as agricultural in 1992 or 1997.

^b Non-agricultural lands are included for informational purposes only; only pasture/forest areas and cropland areas contribute to emissions.

To estimate the “unspecified” and “estimated” amounts of crushed limestone and dolomite applied to agricultural soils, it was assumed that the fractions of “unspecified” and “estimated” production that were applied to agricultural soils in a specific year were equal to the fraction of “specified” production that was applied to agricultural soils in that same

year. In addition, data were not available for 1990, 1992, and 2000 on the fractions of total crushed stone production that were limestone and dolomite, and on the fractions of limestone and dolomite production that were applied to soils. To estimate the 1990 and 1992 data, a set of average fractions were calculated using the 1991 and 1993 data. These average fractions were applied to the quantity of “total crushed stone produced or used” reported for 1990 and 1992 in the 1994 *Minerals Yearbook* (Tepordei 1996). To estimate 2000 data, the 1999 fractions were applied to a 2000 estimate of total crushed stone found in the USGS *Mineral Industry Surveys: Crushed Stone and Sand and Gravel in the First Quarter of 2001* (USGS 2001).

The primary source for limestone and dolomite activity data is the *Minerals Yearbook*, published by the Bureau of Mines through 1994 and by the U.S. Geological Survey from 1995 to the present. In 1994, the “Crushed Stone” chapter in *Minerals Yearbook* began rounding (to the nearest thousand) quantities for total crushed stone produced or used. It then reported revised (rounded) quantities for each of the years from 1990 to 1993. In order to minimize the inconsistencies in the activity data, these revised production numbers have been used in all of the subsequent calculations.

Uncertainty

Uncertainties in the flux estimates for mineral and organic soils result from both the activity data and the carbon stock and adjustment factors. Each of the datasets used in deriving the area estimates has a level of uncertainty that is passed on through the analysis, and the aggregation of data over large areas necessitates a certain degree of generalization. The default IPCC values for mineral soil carbon stocks under native vegetation as well as values for the base, tillage, and input factors represent broad regional averages. Thus, the values have potentially high uncertainty when applied to specific combinations of climate, soil, and

Table 6-14: Quantities of Applied Minerals (Thousand Metric Tons)

Description	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
Limestone	19,012	20,312	17,984	15,609	16,686	17,297	17,479	16,539	14,882	16,894	17,443
Dolomite	2,360	2,618	2,232	1,740	2,264	2,769	2,499	2,989	6,389	3,420	3,531

land management conditions. Similarly, measured carbon loss rates from cultivated organic soils vary by as much as an order of magnitude, depending on climate, land use history and management intensity.

Revised inventory approaches to better quantify uncertainty and to better represent between-year variability in annual fluxes are being developed and are currently under review. A modification of the inventory based on the *Revised 1996 IPCC Guidelines* uses field data specific to the United States to statistically estimate land-use and management factor values and baseline carbon stocks (see Box 6-1). These are combined with other error estimates in a Monte-Carlo simulation to generate 95 percent confidence intervals for average annual fluxes for mineral and organic soils. An annual activity-based inventory using a dynamic simulation model is also being tested (see Box 6-2). The method uses similar climate, soil, and land-use/land-management

databases as the IPCC approach, but is more capable of estimating annual variation in fluxes, and including the effects of long-term trends in agricultural productivity on soil carbon stocks.

Uncertainties in the estimates of emissions from liming result from both the methodology and the activity data. It can take several years for agriculturally-applied limestone and dolomite to degrade completely. The IPCC method assumes that the amount of mineral applied in any year is equal to the amount that degrades in that year, so annual application rates can be used to derive annual emissions. Further research is required to determine actual degradation rates, which would vary with varying soil and climatic conditions. However, application rates are fairly constant over the entire time series, so this assumption may not contribute significantly to overall uncertainty.

Box 6-1: Estimating Uncertainty Using a Revised IPCC Approach

A modification of the IPCC methodology, which incorporates estimates of factor values and baseline carbon stocks based on U.S.-specific data and a Monte Carlo uncertainty analysis, is currently under peer-review (Ogle et al. in prep.). Based on an extensive literature review of more than 1,000 published studies, IPCC factor values and organic soil emission rates have been re-estimated using field studies specific to U.S. conditions. Linear mixed-effect models were used to derive probability density functions (PDF) for each factor value. In addition, PDFs for baseline soil carbon levels were derived from the National Soil Characterization Database, which contains carbon measurements for thousands of soil pedons sampled in the United States (NRCS 1997). Finally, PDFs were derived for the area estimates of individual land-use/management categories based on the expansion factors from the *National Resources Inventory* and tillage management data provided by the Conservation Technology Information Center. The expansion factors are a statistical representation of the land area in the inventory. Probability density functions for the climate/soil/land-use/land-management categories were based on mean and variance estimates for individual land areas, assuming normality, while accounting for the inter-dependence in land use between starting and ending years in the inventory. A Monte Carlo approach (Smith and Heath 2001) was used to estimate overall uncertainty for carbon fluxes associated with each agricultural management activity. The Monte Carlo procedure simulated 50,000 estimates, using an iterative process in which random selections were made from the probability density functions described above. This method provides estimates of carbon flux between U.S. agricultural soils and the atmosphere with statistically valid 95 percent confidence intervals.

Preliminary results suggest that basing the inventory on U.S.-specific data sources gives somewhat lower estimates for carbon sequestration on mineral soils and higher estimates for carbon emissions from cultivated organic soils. Preliminary results show confidence intervals of about ± 45 percent of the mean for mineral soil fluxes and ± 30 percent for organic soil fluxes.

Box 6-2: Century Model Estimates of Soil Carbon Stock Changes on Cropland

Soil carbon stock changes on U.S. cropland were estimated using a dynamic ecosystem simulation model called Century (Metherell et al. 1993, Parton et al. 1994). This method differs from the IPCC approach in that annual changes are computed dynamically as a function of inputs of carbon to soil (i.e., crop residues, manure, and sewage sludge) and soil carbon decomposition rates, which are governed by climate and soil factors as well as management practices. The model simulates all major field crops (maize, wheat and other small grains, soybean, sorghum, cotton) as well as hay and pasture (grass, alfalfa, clover). Management variables included tillage, fertilization, irrigation, drainage, and manure addition.

Input data were the same as that used in the IPCC-based method, (i.e., mean climate variables were from the PRISM database; crop rotation, irrigation and soil characteristics were from the *National Resources Inventory* (NRI); and tillage data were from the Conservation Technology Information Center). Differences with respect to the IPCC-based method were as follows: 1) climate values were applied to each individual MLRA to drive the simulation (as opposed to their use for classification into broad climate zones in the IPCC method) and 2) soil physical parameters, which influence decomposition rates and soil water balance, specific to each MLRA point were used with Century, while, in the IPCC method, soils information were used to group NRI points by broad soil taxonomic classes. In the Century-based analysis, land areas having less than 5 percent of total area in crop production were excluded and several less-dominant crops (e.g., vegetables, sugar beets and sugar cane, potatoes, tobacco, orchards, and vineyards), for which the model has not yet been parameterized, were not included. Thus, the total area included in the Century analysis (149 million hectares) was smaller than the corresponding area of cropland (165 million hectares) included in the IPCC estimates.

Preliminary results using the Century model suggest (as with the IPCC model) that U.S. cropland soils (excluding organic soils) are currently acting as a carbon sink, of about 21 Tg C/year (77 Tg CO₂ Eq./year) (average rates for 1992 through 1997). The main management changes responsible for the increase in mineral soil carbon stocks, according to the Century approach, are the same as those indicated by the IPCC method: reduced tillage intensity; establishment of the Conservation Reserve Program; reduced bare fallow; and some increase in hay area. In addition, the Century analysis includes the effect of increasing residues inputs due to higher productivity on cropland in general, which contributes to the increase in soil carbon stocks.

Potential advantages of a dynamic simulation based approach include the ability to use actual observed weather, observed annual crop yields, and more detailed soils and management information to drive the estimates of soil carbon change. This would facilitate annual estimates of carbon stock changes and CO₂ emissions from soils that would better reflect interannual variability in cropland production and weather influences on carbon cycle processes.

There are several sources of uncertainty in the limestone and dolomite activity data. When reporting data to the USGS (or U.S. Bureau of Mines), some producers do not distinguish between limestone and dolomite. In these cases, data are reported as limestone, so this could lead to an overestimation of limestone and an underestimation of dolomite. In addition, the lack of comprehensive limestone and dolomite end-use data makes it necessary to derive amounts of “unspecified” and “estimated” crushed

limestone and dolomite applied to agricultural soils based on “specified” production data. Lastly, the total quantity of crushed stone listed each year in the *Minerals Yearbook* excludes American Samoa, Guam, Puerto Rico, and the U.S. Virgin Islands. The *Mineral Industry Surveys* further excludes Alaska and Hawaii from its totals.

Changes in Yard Trimming Carbon Stocks in Landfills

As is the case with carbon in landfilled forest products, carbon contained in landfilled yard trimmings can be stored indefinitely. In the United States, yard trimmings (i.e., grass clippings, leaves, branches) comprise a significant portion of the municipal waste stream, and a large fraction of the collected yard trimmings are discarded in landfills. However, both the amount of yard trimmings collected annually and the fraction that is landfilled have declined over the last decade. In 1990, nearly 32 million metric tons (wet weight) of yard trimmings were collected at landfills and transfer stations (Franklin Associates 1999). Since then, programs banning or discouraging disposal have led to an increase in backyard composting and the use of mulching mowers, and a consequent 35 percent decrease in the amount of yard trimmings collected. At the same time, a dramatic increase in the number of municipal composting facilities has reduced the proportion of collected yard trimmings that are discarded in landfills—from 72 percent in 1990 to 36 percent by 2000. The decrease in the yard trimmings landfill disposal rate has resulted in a decrease in the rate of landfill carbon storage from about 19.1 Tg CO₂ Eq. in 1990 to 6.4 Tg CO₂ Eq. in 2000 (see Table 6-15).

Table 6-15: Net CO₂ Flux from Landfilled Yard Trimmings

Year	Tg CO ₂ Eq.
1990	(19.1)
1995	(12.2)
1996	(10.2)
1997	(9.5)
1998	(8.3)
1999	(7.3)
2000	(6.4)

Note: Parentheses indicate net storage. Lightly shaded area indicates values based on projections.

Methodology

The methodology for estimating carbon storage is based on a life-cycle analysis of greenhouse gas emissions and sinks associated with solid waste management (EPA 1998). According to this methodology, carbon storage is the product of the weight of landfilled yard trimmings and a storage factor. The storage factor, which is the ratio of the weight of the carbon that is stored indefinitely to the wet weight of the landfilled yard trimmings, is based on a series of experiments designed to evaluate methane generation and residual organic material in landfills (Barlaz 1998). These experiments analyzed grass, leaves, branches, and other materials, and were designed to promote biodegradation by providing ample moisture and nutrients.

Barlaz (1998) determined carbon storage factors, on a dry weight basis, for each of the three components of yard trimmings: grass, leaves, and branches (see Table 6-16). For purposes of this analysis, these were converted to wet weight basis using assumed moisture contents of 0.6, 0.2, and 0.4, respectively. To develop a weighted average carbon storage factor, the composition of yard trimmings was assumed to consist of 50 percent grass clippings, 25 percent leaves, and 25 percent branches on a wet weight basis. The weighted average carbon storage factor is 0.23 (weight of carbon stored indefinitely per unit weight of wet yard trimmings).

Table 6-16: Storage Factor (kg C/kg dry yard trimmings) Moisture Content (kg water/kg wet yard trimmings), Composition (percent) and Converted Storage Factor (kg C/kg wet yard trimmings) of Landfilled Yard Trimmings

Component	Grass	Leaves	Branches
Storage Factor ^a	0.32	0.54	0.38
Moisture Content	0.60	0.20	0.40
Composition	50%	25%	25%
Converted Storage Factor ^b	0.13	0.43	0.23 ^c

^a From Barlaz (1998)
^b The converted storage factor for each component is the product of the original storage factor and one minus the moisture content; the weighted average storage factor is obtained by weighting the component storage factors by the composition percents.
^c Value is also value of weighted average.

Data Sources

The yard trimmings discards data were taken from the report *Characterization of Municipal Solid Waste in the United States: 1998 Update* (Franklin Associates 1999), which provides estimates for 1990 through 1997 and forecasts for 2000 and 2005 (Table 6-17). Yard trimmings discards for 1998 through 2000 were projected using the Franklin Associates (1999) forecast of generation and recovery rates (i.e., decrease of 6 percent per year, increase of 8 percent per year, respectively) for 1998 through 2000. This report does not subdivide discards of individual materials into volumes landfilled and combusted, although it does provide an estimate of the overall distribution of solid waste between these two management methods (i.e., ranging from 81 percent and 19 percent respectively in 1990, to 76 percent and 24 percent in 2000) for the waste stream as a whole.¹⁰ Thus, yard trimmings disposal to landfills is the product of the quantity discarded and the proportion of discards managed in landfills. As discussed above, the carbon storage factor was derived from the results of Barlaz (1998), and assumed moisture contents and component fractions for yard trimmings.

Uncertainty

The principal source of uncertainty for the landfill carbon storage estimates stems from an incomplete understanding of the long-term fate of carbon in landfill environments. Although there is ample field evidence that many landfilled organic materials remain virtually intact for long periods, the quantitative basis for predicting long-term storage is based on limited laboratory results under experimental conditions.¹¹ In reality, there is likely to be considerable heterogeneity in storage rates, based on 1) actual composition of yard trimmings (e.g., oak leaves decompose more slowly than grass clippings) and 2) landfill characteristics (e.g., availability of moisture, nitrogen, phosphorus, etc.). Other sources of uncertainty include the estimates of yard trimmings disposal rates, which are based on extrapolations of waste composition surveys, and the extrapolation of values for 1998 through 2000 disposal from estimates for the period from 1990 through 1997.

Table 6-17: Collection and Destination of Yard Trimmings (Million Metric Tons, wet)

Destination	1990		1995	1996	1997	1998	1999	2000
Municipal Composting Facilities	3.8		8.2	9.4	10.4	10.6	10.8	10.9
Discarded	27.9		27.4	26.9	23.9	13.0	11.4	10.0
Landfill	22.8		22.2	21.7	19.2	17.1	14.5	7.6
Incineration	5.2		4.3	3.8	3.5	3.1	2.7	2.4
Total	31.8		26.9	25.3	25.2	23.6	22.2	20.9

Note: Lightly shaded area indicates values based on projections.

¹⁰ These percents represent the percent of total MSW discards, after recovery for recycling or composting.

¹¹ In addition, there was a mass balance problem with the experimental results that are used here to derive the yard trimmings carbon storage factor. In particular, the carbon storage factor for leaves that was determined experimentally by Barlaz (1998) was greater than Barlaz’s measured carbon content of the leaves.